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ABSTRACT

The sequestration of excess atmospheric C into resilient and long-lasting belowground pools is of increasing global importance to mitigate greenhouse gas emissions. One land use that is particularly amenable to this type of manipulation is agricultural land, which often has high sequestration potential. However, this capacity can depend on local factors such as cropping history, soil type and climate. Critically, it may also be limited by N availability. In this study we used the retrospective (repeated measures) methodology to assess the impact on previously arable land of land use change by either afforestation with two species of broadleaf trees planted at 800 or 1600 stems ha^{-1} (T800 and T1600), or reversion to rough grassland (NT) for 21 years. We quantified the concentration, distribution and total stocks of organic C and N in the upper 0-30 cm of a common soil type found in NE Scotland and investigated the robustness of C:N to land use change. Finally we estimated ecosystem stocks of C and N, and how these were partitioned between plant and soil components. We found increases in the overall concentrations of soil C from 4.6% to 5.8%, and N from 0.32% to 0.43%. The increase in soil C stocks over the experiment was in the order NT > T800 > T1600 and each treatment differed significantly. The same pattern was seen for increases in N stocks but here the increases in NT and T800 were significantly greater than for T1600. Overall, stocks were higher in the rough grassland plots than under trees by 35 Mg ha^{-1} for C and 2.2 Mg ha^{-1} for N. These increases in stocks were accompanied by a highly significant narrowing in C:N with time across all treatments from 14.6 to 13.6 and differences seen between upper and lower soil layers in 1991 had disappeared by 2012. From an average of 151 Mg ha^{-1} in 1991, the system C stock (soil + plants) had increased to between 202 Mg ha^{-1} (NT) and 221 Mg ha^{-1} (T1600) by 2012, with between 96% (NT) and 73% (T1600) of the C in the soil. Concurrently the system stock of N had increased to between 14.1 Mg ha^{-1} (NT) and 11.3 Mg ha^{-1} (T1600), from an average of 9.4 Mg ha^{-1} in 1991, with between 99% (NT)

and 81% (T1600) of the system N in the soil. Although in 2012 there were significantly greater soil C stocks in NT, this was offset by C accumulation in the treatments containing trees, such that overall there were no treatment differences. However, this was not seen with system N stocks in 2012, which were significantly larger in the NT treatment than in those with trees. Mean annual rates of C and N accumulation in the systems were greatest in NT (2.7 and 0.22 Mg ha^{-1} yr^{-1}) and least in T1600 (0.7 and 0.08 Mg ha^{-1} yr^{-1}). These results are important in the context of land use strategies aimed at pollution mitigation, such as C sequestration and nitrate leaching. They are also relevant to the possible effects of land use change, especially reversion to agricultural use, of land previously taken out of production.

KEYWORDS

Afforestation, Abandonment, Sequestration, Climate change, Greenhouse gas mitigation, Temperate

1. INTRODUCTION

The dynamics of soil organic matter (SOM) continues to represent an active and topical area of research due to the central role it is known to play in ecosystem functioning. As the dominant constituent of SOM, carbon represents one of the most commonly determined of soil attributes with concentrations ranging widely from representing only a minor (<1.0%) to a major (>45%) component of surface soils. At any particular time SOM will consist of a diverse heterogeneous mix of compounds with respect to their origin, composition, reactivity and importantly lability (biological availability) and therefore persistence at any particular location. The distribution and total quantities of SOM present within a soil profile reflect a combination of background attributes (*e.g*., climatic and edaphic) together with more site-specific 'management' factors (*e.g*., drainage and cropping histories).

Wide-scale and historical changes in land use have therefore had dramatic influences on the quantity and quality of SOM. Typically, although not exclusively, for temperate regions a change from 'native' to 'agricultural' use has resulted in a loss of SOM (McLauchlan, 2006), which in arable systems can also be associated with a physical redistribution within the soil profile as a result of cultivation and soil mixing (Nierop *et al*., 2001). An additional consequence of changes to land use can be changes in the elemental composition, type and reactivity of organic compounds. Land use changes that include cultivation, drainage, liming and farming system have dramatically influenced the capacity of soil to act as a sink for C. Kirk and Bellamy (2010) suggested that for the UK, past changes in land use were better able to explain C turnover than recent changes in climate (temperature or moisture). A combination of interacting local factors which broadly equate to the 'soil formation factors' (defined by Jenny, 1941) determine the balance between accumulation and mineralisation of organic matter recently defined as an 'ecosystem property' by Schmidt *et al.*, (2011).

The reversibility of the loss of SOM in agricultural soils by the cessation of ploughing is well established (*e.g.* Freibauer *et al*., 2004). It is well demonstrated by the classical long-term studies at Rothamsted (Poulton *et al*., 2003), which showed the accumulation of SOM and organic C after the removal of arable land from agricultural production (the so-called 'wilderness area'). Interestingly, this study also reported on accompanying changes in total soil N. Reversion of old arable land to woodland resulted in a widening of the C:N ratio as a result of an increase in the quantity of SOC (doubling over 100 years) which was accompanied by a smaller increase in N. The authors suggested that the presence of sufficient 'biologically available' N was a key factor influencing this soil's ability to retain and therefore accumulate SOM. The role that stoichiometric relations between the major nutrients (C:N), which are well-constrained at the global scale, might have with respect to their regulatory influences upon SOM dynamics and average atomic C:N ratios is becoming increasingly recognised (e.g. Cleveland and Liptzin, 2007; Zhao et al., 2015a).Similarly, in abandoned agricultural field soils the availability of N appears to control the accumulation of C in abandoned agricultural soils (Knops and Tilman, 2000).

Changing land use influences not only the quantity of C and N likely to be stored in ecosystems but can also influence the physical distribution between above and below ground pools. Where land use changes are substantial, such as those that accompany afforestation of arable land, then the distribution and potential medium term storage of C and N can vary over time. This situation has been highlighted by Sharrow and Ismail (2004) where pastures stored > 90% of their C and N below ground while coniferous plantations had the bulk stored in above ground and litter layers. These authors hypothesized that silvopastoral systems should accumulate more C and N than either grass or tress grown alone. Over the first 11 years these agroforestry plots, located in western Oregon, accumulated 740 and 520 kg ha^{-1} yr^{-1} of C more than forest and pasture respectively, however the N story was not quite so clear.

Here our objective is to quantify the effect of a change in land use from long-term arable into either unmanaged rough grassland or low-density deciduous tree planting after a period of 21 years on the concentration, distribution and total stocks of organic C and N in the upper 0-30 cm of a common soil type found in NE Scotland. In particular we investigate how robust the ratio of C:N is to land use change and the extent to which it may differ between these land uses. Finally we estimate ecosystem stocks of C and N, and how these are partitioned between plant and soil components. Results are considered in relation to land use change strategies aimed at pollution mitigation, such as C sequestration and nitrate leaching.

2. METHODS

2.1. **Experimental Design**

The experiment was established in 1991 at Craibstone Estate, Aberdeen in NE Scotland and basic site characteristics are given in Table 1. The soil is a Dystric cambisol belonging to the Countesswells Association and Countesswells/Terryvale Series (Glentworth and Muir, 1963). It is derived from granitic parent material and imperfectly drained due in part to a semi-continuous indurated (low permeability) soil horizon at a depth of 35-40 cm. The land had previously been in arable cropping for at least 20 years and probably much longer. In each of three sub-blocks, small saplings of sycamore (*Acer pseudoplatanus* L.) and wild cherry (*Prunus avium* L.) were planted by hand at either 800 (T800) or 1600 (T1600) stems ha^{-1} into four plots of 30m x 37.5m, plus a 9 m x 60 m area that was not planted (no tree, NT), all surrounded by a 5 m wide guard strip of grass. For the first six years of the experiment each sub-block of four tree + NT plots had one of three different understoreys (ryegrass, white clover or vegetation free), which together formed one experimental block (15 plots). Spatial layout was randomised within blocks, which were replicated three times (45 plots). The understorey treatments were not maintained beyond six years and reverted to a cover of rough grassland (NT plots) or an increasingly negligible cover of rough grassland/leaf litter (T

plots) for the remaining 15 years. Both understorey and trees were not managed during this period although they were subject to low levels of grazing and browsing by wild animals. Meteorological data were obtained from a weather station at Craibstone, about 0.5 km from the study site (

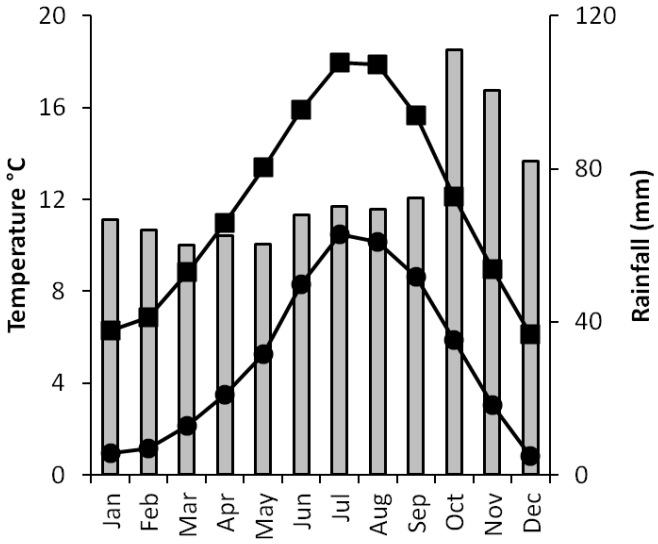

Figure 1).

2.2. **Sampling and Analyses**

Before the trees were planted in spring 1991 and again in spring 2012, 10 soil cores (3.5 cm diameter) were taken per plot after removal of any surface vegetation, split into two depth layers (0-15 cm and 15-30 cm) and pooled by layer to give 2 samples per plot. The upper layer is approximately the same depth as the former plough depth (Ap horizon). Cores were taken at random locations excluding the outer 2 m of plots and always on the midline between tree rows. Even after 21 years there was no measurable organic layer at the site. For determination of bulk density before planting, bulk density cores were taken from the middle of each sampling layer from 8 locations across the site. In 2012, one bulk density core (94.3 cm^3) was taken from the middle of each sampling layer in each plot from a random location excluding a 2 m perimeter. Soils were air dried, sieved <2 mm, dried in a forced-air oven at

60°C for 48 h and ball milled. Total C and N concentrations in soil samples were analysed on a continuous flow ANCA-NT gas/solid/liquid preparation module coupled to a "TracerMass" mass spectrometer (Europa Scientific, UK). Archived samples from 1991 were analysed at the same time as those from 2012, to remove bias due to differences between analytical systems.

2.3. **Vegetation Data**

Diameter at breast height (DBH) was measured for 10 trees in each plot, selected at random with the exclusion of a border two stems deep around the edge of the plot, in early May 2015. Tree stem dry mass (trunk + branches) was estimated from the DBH data using allometric equations (Bunce, 1968): Ln[biomass (kg)] = a + b.ln[DBH (cm)] using values for sycamore of a = -5.5705, b = 2.5294; cherry a = -5.4453, b = 2.5051. These values for cherry are the "combined" figures of ash, birch, oak and sycamore from Bunce (1968) and used by Poulton *et al*. (2003), as no species-specific values are available.

Stem dry mass was used in preference to total (stem + leaf) dry mass due to the transient nature of leaf cover in broadleaf species and their relatively small contribution to overall total dry mass in trees of the age studied here. Root dry mass was calculated according to Enquist and Niklas (2002): $Ms = 2.61Mr^{1.1}$ where Ms = stem (branch + trunk) dry mass (kg) and Mr = root dry mass (kg). All figures were converted into C contents assuming a C content of 50% (Matthews, 1993) and reduced to account for the two-year difference in tree and soil sampling dates by linear correction Poulton *et al*. (2003). This used a linear regression derived from the Carbon Lookup Tables (West and Matthews, 2011) for standing mass of C per year in a mixed deciduous stand (category SAB) of yield class 4 and 2.5 m tree spacing, aged 15 – 25 years: standing C = 5.184.year - 68.624, $R^2 = 0.9978$. Tree N contents were calculated using values for "birch + other deciduous" (Kauppi *et al*., 1995): Branch dry mass

was allocated as 15% of stem dry mass and N concentrations were taken as 0.12% (trunk), 0.5% (branches) and 0.3% (roots).

For the rough grassland plots, biomass was estimated from the mean yield of hay plots (4.33 Mg ha^{-1}) from a long-term crop rotations trial within 0.5 km of the study site over the same time period that had received only one application of farmyard manure every 7 years since 1922 (Walker *et al*., 2010). Root biomass was calculated using the root:shoot ratio of 2.73, aboveground N of 1.38% and root N of 0.96%. These values were taken from the grass understorey plots in the initial years of the current trial when the trees were small (mean heights: cherry 44.6 cm, sycamore 50.4 cm).

2.4. **Statistical Analysis**

This is an example of a retrospective design, which re-samples the same plots over a given period of time (repeated measurements). It is usually considered the most powerful and least biased design, since it eliminates the variation of error associated with different conditions between sites in paired-site and chronosequence designs (Laganière *et al*., 2010). Repeated measures analyses of variance were performed for a "factorial plus control" design with tree species and planting density being the factorial elements, and the unplanted area the control (Payne, 2012). Density × species was nested within planted × depth for all assessed variables except bulk density, as this was not measured on a plot basis at the start of the experiment. Sphericity was not assumed for the repeated measures analyses and corrections were based on values of Geisser-Greenhouse epsilon. An initial analysis showed that after 21 years there were no significant effects of the original understorey treatments, so these were included in the analysis as sub blocks within the overall blocking structure.

All analyses used GenStat Release 16.1 64-bit (VSN International Ltd.) following checks on the normality of the data and homogeneity of variances. The statistical significance of differences between individual means was assessed by LSD at a significance level of 0.05.

3. RESULTS

3.1. **Composition of soil C and N, and BD**

In the topsoil layer 0-30 cm for all treatments, the concentration of soil C increased significantly ($P<0.001$) over 21 years from 4.6 ± 0.1% to 5.8 ± 0.1%, and there was a significantly higher C concentration in the upper layer than the lower (5.5 ± 0.1% vs. 4.8 ± 0.1%, $P<0.001$). The increase in C concentration with time was significant at the lower depth for all treatments, but in the upper depth was only significant at T800 (Table 2). By 2012 the difference in C concentration between the two depths had become non-significant in all treatments.

The pattern of changes of soil N concentration was very similar to those of C concentration. In the topsoil layer 0-30 cm for all treatments, there was a significant ($P<0.001$) increase in N concentration with time from 0.32 ± 0.01% to 0.43 ± 0.01%, and a significantly higher N concentration in the upper layer than the lower (0.40 ± 0.01% vs. 0.34 ± 0.01%, $P<0.001$). While there was a significantly higher N concentration at the lower depth than the upper depth in all treatments in 1991, this had disappeared by 2012 (Table 2).

In 1991 there was no significant difference in the bulk density of the soil with depth, but by 2012 the bulk density of the lower layer was significantly greater ($P<0.001$) than that in the upper layer in all treatments. (Table 2). Overall, there was also a significant effect of tree planting on the lower layer, with T1600 having a lower bulk density than NT in 2012. However, within each soil layer in each treatment there were no significant changes in bulk density with time.

Tree species had no significant effect on concentrations of C or N over time (data not shown). Overall, sycamore plots tended to have slightly lower values at the start of the experiment and stayed that way.

3.2. **Change in soil C and N stocks over time**

Total soil C and N stocks (0 to 30 cm depth) increased significantly ($P<0.001$) with time in all treatments. In 2012, C and N stocks in NT were significantly ($P<0.05$) greater than in T800 and T1600, whose C and N stocks did not differ significantly between treatments (Table 3). As with the concentration data, tree species had no significant effect on stocks of C or N over time (data not shown).

The stocks of soil C and N increased significantly ($P<0.001$) at the lower soil depth in all treatments with time (Table 3). Stocks of both in the upper layer showed an increasing trend in all treatments but these increases were significant only in T800. At the start of the experiment there was no significant difference between C and N stocks at the two depths in all treatments. By 2012 the stocks of both C and N in NT were significantly greater in the lower layer than in the upper (Table 3).

3.3. **Change in soil organic matter quality (C:N)**

Over the experiment there was a highly significant ($P<0.001$) narrowing in C:N with time across all treatments from 14.6 ± 0.1 to 13.6 ± 0.1. This pattern was also seen in all treatments at both depths (Table 4). The mean change in C:N over time was -0.7 ± 0.1 for the upper layer and -1.2 ± 0.1 for the lower layer.

In 1991 C:N was significantly wider at depth within each treatment. By 2012 these differences had disappeared, when there were no significant differences between the two layers within treatments (Table 4). Across all treatment/depth combinations in 2012 there were no significant differences in C:N except in one case (T800, 0-15 cm).

3.4. **Change in system C and N stocks over time**

The increase in soil C stocks over the experiment was in the order NT > T800 > T1600 and each treatment differed significantly (Table 5). The same pattern was seen for increases in N stocks but here the increases in NT and T800 were significantly greater than for T1600.

Although the increases in system C (soil + vegetation) were in the same order as for the increases in soil C stocks, there were no significant differences between treatments. Similarly, the increases in system N stocks did not differ significantly between treatments (Table 5). Mean annual rates of C and N accumulation were greatest in NT (2.7 and 0.22 Mg ha^{-1} yr^{-1}) and least in T1600 (0.7 and 0.08 Mg ha^{-1} yr^{-1}).

By 2012 the system C stock had increased to between 202 ± 9 Mg ha^{-1} (NT) and 221 ± 6 Mg ha^{-1} (T1600), from an average of 151 ± 3 Mg ha^{-1} in 1991 (Figure 2). Between 96% (NT) and 73% (T1600) of the system C was in the soil. Concurrently the system stock of N had increased to between 14.1 ± 0.6 Mg ha^{-1} (NT) and 11.3 ± 0.4 Mg ha^{-1} (T1600), from an average of 9.4 ± 0.2 Mg ha^{-1} in 1991, with between 99% (NT) and 81% (T1600) of the system N in the soil (Figure 2). Although in 2012 there were significantly greater soil C stocks in NT, this was offset by vegetation C accumulation in the treatments containing trees, such that overall there were no treatment differences. However, this was not seen with system N stocks in 2012, which were significantly larger in the NT treatment than in those with trees (Figure 2).

4. DISCUSSION

4.1. **Accumulation of soil C and N**

The large accumulation of soil C stocks over 21 years of between 14.0 and 33.9 Mg ha^{-1} (0.67 and 1.61 Mg ha^{-1} yr^{-1}) are somewhat greater than in some previous studies that considered afforestation. Soil C accumulation has been reported as 63.7 Mg ha^{-1} (0.55 Mg ha^{-1} yr^{-1}) 120 years after reversion to woodland in SE England (Poulton *et al*., 2003); up to 37 Mg ha^{-1} over about 100 years (0.37 Mg ha^{-1} yr^{-1}) in the soils of western New England (Clark and Johnson, 2011) and 21 Mg ha^{-1} in six sites across Europe (Poeplau and Don, 2013). While caution must be exercised in the comparison of rates of accumulation drawn from different time

periods as accumulation trajectories tend to be non-linear (Poeplau *et al*., 2011), they are useful in setting the general context for possible rates in different systems.

The % change in soil C stocks reported here (+26% in T800 and +10% in T1600) agree well with the ranges reported in a meta-analysis of SOC accumulation following afforestation (Laganière *et al*., 2010). However, our results contradict the findings of a study from Denmark which concluded that afforestation of former cropland within the temperate region may induce soil C loss during the first decades (Bárcena *et al*., 2014).

Our study was planted at relatively low densities and used tree species considered low-yielding compared with commercial afforestation (West and Matthews, 2011), and thus may be considered more similar to some agroforestry systems. The rates of change in C stocks reported above are in general agreement with recent studies of the C sequestration potential of agroforestry systems in European agriculture of 1.5 – 4 Mg ha^{-1} yr^{-1} (Hamon *et al*., 2009) and 2.75 Mg ha^{-1} yr^{-1} (Aertsens *et al*., 2013).

The accumulation of 55.9 Mg ha^{-1} (2.66 Mg ha^{-1} yr^{-1}) in previously arable soils left to revert to rough grassland (NT plots) is much greater than previously reported in most studies, although Rees *et al.* (2005) reported annual root-derived inputs up to 2.8 Mg ha^{-1} of C from perennial ryegrass. A 10 year old successional system following cropping in Michigan showed an increase in total soil C (0-50 cm) at a rate of 0.79 Mg ha^{-1} yr^{-1} (DeGryze *et al*., 2004) and a study of six sites across Europe found a mean increase in SOC stock after land use change from cropland to grassland of 18 Mg ha^{-1} (Poeplau and Don, 2013). Interestingly, a recent modelling prediction of change in soil C stocks indicated that an accumulation rate of C associated with a change in land use from arable to grassland for this area of Scotland could be in the order of 4 Mg ha^{-1} yr^{-1} over a 10 year period (Smith *et al*., 2009).

That grassland soils may accumulate more soil C than woodlands has been shown in a number of studies (Franzluebbers *et al*., 2000; Garten and Ashwood, 2002). This has been

attributed to greater input of root C in grasslands than forests (Laganière *et al*., 2010), due to herbaceous plants allocating a much higher proportion of biomass to root systems (Bolinder *et al*., 1997) and having a generally faster turnover of roots than trees (Kuzyakov and Domanski, 2000; Guo *et al*., 2007).

Concomitant with the large increases in soil C stocks were increases in N stocks of between 1.7 Mg ha^{-1} (83 kg ha^{-1} yr^{-1}) in T1600 to 4.7 Mg ha^{-1} (221 kg ha^{-1} yr^{-1}) in NT. While a trend for increasing soil N stocks with afforestation or reversion to rough grassland has been reported (Knops and Tilman, 2000; Shi *et al*., 2015), to our knowledge none have found such large accumulations. A soil N accumulation of 13 kg ha^{-1} $yr^{-1}$was reported for soils (to a depth of 55 cm) formerly used for cultivated crops in western New England (Clark and Johnson, 2011), and Poulton *et al*. (2003) reported a maximum rate of N accumulation of 59 kg ha^{-1} yr^{-1} in South East England (for tree + soil, although the tree component was small).

The sampling depth used in this study was based on an extensive site survey before the experiment was established. This revealed an average soil depth over the site of 35 cm above an induration horizon. Thus the changes in stocks reported to a depth of 30 cm are likely to be a close representation of actual changes through the profile. The greatest increase in soil C stocks occurred in the 15-30 cm depth, suggesting that perhaps majority of the increase comes from belowground production rather than the input of litter and aboveground material. However many studies have reported the opposite, with soil C stocks increasing most within the upper soil layer (Davis and Condron, 2002; Foote and Grogan, 2010; Korkanc, 2014).

4.2. **Change in composition of SOM**

Prior to the beginning of the experiment the field was managed as part of an arable rotation and therefore subjected to repeated cultivation and mixing/aeration of the topsoil for a minimum twenty year period. The organic matter present in topsoil samples collected in 1991 should thus reflect an extended time period that has experienced regular inputs of a wide

range of crop residues, lime and fertiliser/manure, providing general soil conditions likely to favour mineralisation processes. The intense period of microbial activity immediately after ploughing and cultivation has been regularly reported and evidenced in the vicinity of our site by significant leaching of dissolved organic carbon and nitrate immediately after ploughing of an adjacent field (Green, 1997). Hutchinson *et al*. (2007) suggested that in temperate regions 20-30% of the organic carbon in the top 30 cm of soil is susceptible to rapid loss due to frequent cultivation. The organic matter content and composition present in 1991 will more closely reflect the dynamic situation reflecting 'local arable' conditions which would have favoured a lower organic matter content than semi-natural soil.

The post-1991 change in land use (with associated changes in management) would have shifted the soil system to conditions that are likely to favour accumulation of organic matter. Interestingly, while both C and N concentrations and soil stocks have increased, the relative rate of increase in N has been greater (Table 4). As a consequence there has been a significant narrowing of the measured SOM C:N in both soil depths and all treatments. This suggests that the "fresh" SOM was composed of more chemically stable C-rich compounds is true, then the organic material that has accumulated over the measurement period would have had to have an even narrower C:N ratio.

4.3. **System accumulation of C and N**

If it is assumed that soil C stock is directly proportional to the annual plant C input (Jenkinson *et al*., 1992), the higher soil C stocks for NT indicate that substantially more C is returned to soil in this treatment than in those planted with trees. This concurs with two recent studies, which found that silvopastoral systems accumulated significantly more soil C than systems more densely planted with trees (Baah-Acheamfour *et al*., 2015; Hamon *et al*., 2009). However, when such comparisons include the C stored in tree biomass then, as in the

present study, the differences between systems' soil C stocks are negated or those with more trees may have higher C stocks (Beckert *et al*., 2015). We found no differences in system C or N stocks between the different tree species. The lack of country-specific allometric coefficients for the biomass of cherry meant that we used the combined value from a range of common British broadleaves species. Although Bunce (1968) regarded this as giving an acceptable level of accuracy for biomass estimates, it has the effect of possibly reducing any difference between species in the current study. However, the lack of any significant differences attributable to tree species in any of the other variables we measured supports this suggestion that there was little difference between them.

Mean ecosystem C accumulation rates of 3.6 Mg ha^{-1} yr^{-1} reported here agree closely with those found in Denmark of 2.2 Mg ha^{-1} yr^{-1} (Bárcena *et al*., 2014) and 2.4 to 4.5 Mg ha^{-1} yr^{-1} (Vesterdal *et al*., 2002). They are somewhat lower than found in Spanish grasslands planted with highly productive eucalypt and pine stands, which had ecosystem C accumulation rates of up to 11 Mg ha^{-1} yr^{-1} (Perez-Cruzado *et al*., 2012). The proportion of system C allocated to trees (21% in T800 to 33% in T1600) is lower than the 55% reported by Bárcena *et al*. (2014) (2014) in Denmark, but this was for a 42 year old system measured to a depth of 50 cm.

Although individual annual rates for system C and N accumulation cannot be calculated, even if an unlikely linear rate between the two dates is chosen then these mean rates are considerable: C increased by 3.1 to 3.9 Mg ha^{-1} yr^{-1} and N by 96 to 222 kg ha^{-1} yr^{-1}. While it is easy to see how C is readily supplied from the atmosphere, the source of such large quantities of N is more problematic. Even allowing for a modest residual fertiliser effect from the previous arable production at the study site, and taking into account the atmospheric deposition of N, currently *c*.20 kg ha^{-1} yr^{-1} for Aberdeenshire (DEFRA, 2016), and some free fixation of N (perhaps 1-2 kg ha^{-1} yr^{-1}), then a source for 60 to 180 kg ha^{-1} yr^{-1} of N is still required. Such a shortfall is not a new finding: Confronted by the same question when they

could account for only 20-30 kg ha^{-1} yr^{-1} of the measured 59.3 kg ha^{-1} yr^{-1} of N taken up by the system, Poulton *et al*. (2003) stated that "somehow Broadbalk wilderness managed to acquire twice as much during its early years". One possibility on our site is that the shallow soil and impermeable indurated layer leads to a flow of water across the site that could transport readily soluble N from the greater catchment, which is feasible given that the site is on a gentle slope. Such a level of N accumulation is not an unreasonable possibility given that buffer strips of native vegetation have been reported to increase total soil N by almost 100% (Perez-Suarez *et al*., 2014) and typical losses of nitrate from arable land in the study area are 20-30 kg ha^{-1} yr^{-1} of N (Edwards *et al*., 1990).

Our results indicated that as much C and N was accumulated in the rough grassland system as in the systems planted with trees over 21 years. This has implications for designing systems for the purpose of sequestering these elements, for example C for climate change mitigation or N as a means of reducing quantities lost to watercourses from agricultural systems (buffer strips). Tree- or shrub-based systems are frequently advocated for sequestering both C and N (Laganière *et al*., 2010; Nisbet *et al*., 2011; Powlson *et al*., 2011) but our results suggest that in some situations allowing the land to revert to rough grassland might achieve the same outcome and with lower establishment costs. The large amounts of C and N sequestered also have implications for the longer-term management of these systems. Just as they may be useful for sequestration of C and N when they are growing, care must be taken not to release these back to the environment with future changes in land use.

5. CONCLUSIONS

Soil stocks of C and N increased 21 years after land use change from an arable system, and these increases were greater for land reverting to rough grassland rather than planted with trees. Differential rates of accumulation with depth resulted in a more homogenous distribution of C and N in the soil profile over time. Given that there were no significant

changes in bulk density within layers, the observed increases in C and N stocks represent an increase in SOM. The narrowing of soil C:N seen in all land uses over time suggests that SOM became relatively enriched in N and the initially wider C:N in the upper layer became more evenly distributed over time. Despite these large changes to the systems, the final C:N exhibited remarkably robust stoichiometry across land uses of 13.6.

Although the amount of C accumulated in soil was greatest under rough grassland, these differences disappeared when standing C was taken into account due to the large amounts of C stored in trees. Average annual rates of accumulation of both C and N were high and while C is readily available in the atmosphere the source of up to 222 kg ha^{-1} yr^{-1} of N is unresolved.

All systems accumulated large amounts of C and N. The future land useof these systems must therefore consider the potentially large losses of both C and N if negative environmental consequences are to be avoided and genuine sequestration of C is to be achieved.

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Location	57°11.25' N 2°13.54' W
Altitude (m)	110-130
Attitude	NE
Slope (%)	0.5
pH	5.6
Soil texture	Sandy loam
Parent material	Granite

Table 1. Site characteristics.

Treatment	NT		T800		T1600	
Depth (cm)	0-15	15-30	0-15	15-30	0-15	15-30
Carbon (%)						
1991	5.3Ac ± 0.3	4.0Aa ± 0.1	4.8Abc ± 0.3	4.1Aa ± 0.2	5.1Ac ± 0.3	4.3Aab ± 0.2
2012	6.1Ab ± 0.4	6.0Bab ± 0.6	6.1Bb ± 0.4	5.7Bab ± 0.4	5.8Aab ± 0.5	5.3Ba ± 0.2
Nitrogen (%)						
1991	0.37Ab ± 0.03	0.27Aa ± 0.01	0.34Ab ± 0.02	0.28Aa ± 0.02	0.36Ab ± 0.02	0.28Aa ± 0.01
2012	0.44Ab ± 0.03	0.43Aab ± 0.04	0.46Bb ± 0.03	0.42Bab ± 0.03	0.43Bab ± 0.03	0.38Ba ± 0.02
Bulk density (g cm^{-3})						
1991	0.92Aa ± 0.03	1.08Aa ± 0.04	0.92Aa ± 0.03	1.08Aa ± 0.04	0.92Aa ± 0.03	1.08Aa ± 0.04
2012	0.91Aa ± 0.04	1.26Ab ± 0.01	0.90Aa ± 0.04	1.06Ab ± 0.08	0.86Aa ± 0.03	1.00Ab ± 0.05

Table 2. Soil N and C concentrations and bulk densities for plots that were either unplanted or planted with trees at two densities.

Similar upper case letters indicate no significant change ($P \geq 0.05$) with time within each treatment. Similar lowercase letters indicate no significant difference ($P \geq 0.05$) within each year. Values are ± standard error, n = 9 (no tree), 18 (tree) except for 1991 BD n = 4).

Treatment	NT			T800			T1600		
Depth (cm)	0-15	15-30	0-30	0-15	15-30	0-30	0-15	15-30	0-30
Carbon (Mg ha^{-1})									
1991	72.8Aa ± 3.9	65.3Aa ± 2.3	138.1Xe ± 3.6	65.5Aa ± 3.5	65.8Aa ± 3.9	131.3Xe ± 6.5	70.4Aa ± 3.5	69.2Aa ± 3.1	139.7Xe ± 6.0
2012	83.1Aa ± 7.0	110.9Bb ± 10.7	193.9Yx ± 14.4	79.8Ba ± 4.8	85.3Ba ± 3.3	165.1Yy ± 6.4	74.0Aa ± 5.8	79.7Ba ± 4.5	153.6Yy ± 7.5
Nitrogen (Mg ha^{-1})									
1991	5.12Aa ± 0.34	4.33Aa ± 0.16	9.45Xe ± 0.32	4.65Aa ± 0.23	4.50Aa ± 0.26	9.15Xe ± 0.40	4.98Aa ± 0.24	4.60Aa ± 0.19	9.58Xe ± 0.36
2012	6.07Aa ± 0.52	8.04Bb ± 0.86	14.10Yx ± 1.15	6.06Ba ± 0.37	6.34Ba ± 0.25	12.40Yy ± 0.46	5.52Aa ± 0.43	5.80Ba ± 0.40	11.32Yy ± 0.60

Table 3. Soil N and C stocks (Mg ha^{-1}) over 21 years for plots that were either unplanted or planted with trees at two different densities .

Similar upper case letters indicate no significant change (P≥ 0.05) with time within each treatment. Similar lowercase letters indicate no significant difference (P≥ 0.05) between depths within each variable, except for data in Total columns, where they indicate no significant difference (P≥ 0.05) within each year. All values are Mg ha^{-1} ± standard error, n = 9 (no tree), 18 (tree).

Treatment		NT		T800		T1600	
Depth (cm)		0-15	15-30	0-15	15-30	0-15	15-30
Year	1991	14.4Aab ± 0.4	15.1Ac ± 0.3	14.1Aa ± 0.2	14.6Abc ± 0.3	14.2Aa ± 0.2	15.0Ac ± 0.2
	2012	13.7Bab ± 0.2	13.9Bab ± 0.2	13.2Ba ± 0.2	13.5Bab ± 0.2	13.4Bab ± 0.2	13.9Bb ± 0.2
Change		-0.7	-1.2	-0.9	-1.1	-0.8	-1.1

Table 4. Soil C:N over 21 years for plots that were either unplanted or planted with trees at two different densities.

Similar upper case letters indicate no sig change ($P \geq 0.05$) with time within each treatment × depth combination. Similar lowercase letters indicate no significant difference ($P \geq 0.05$) within each year. Values are ± standard error, n = 9 (no tree), 18 (tree).

Treatment	NT	T800	T1600
Soil C	+55.9a ± 13.6 (2.7)	+33.9b ± 8.0 (1.6)	+14.0c ± 9.2 (0.7)
Soil N	+4.65a ± 1.11 (0.22)	+3.24a ± 0.64 (0.15)	+1.74b ± 0.68 (0.08)
System C	+64.0a ± 14.4 (3.05)	+78.8a ± 8.2 (3.75)	+81.2a ± 9.9 (3.87)
System N	+4.67a ± 1.15 (0.22)	+3.43a ± 0.46 (0.16)	+2.02a ± 0.60 (0.10)

Table 5. Changes in soil and system C and N stocks over 21 years.

Similar letters indicate no significant difference ($P \geq 0.05$) between stocks within each variable. Values are Mg ha^{-1} ± standard error, n = 9 (no tree), 18 (tree). Values in parentheses are mean annual rates of change (Mg ha^{-1} yr^{-1}).

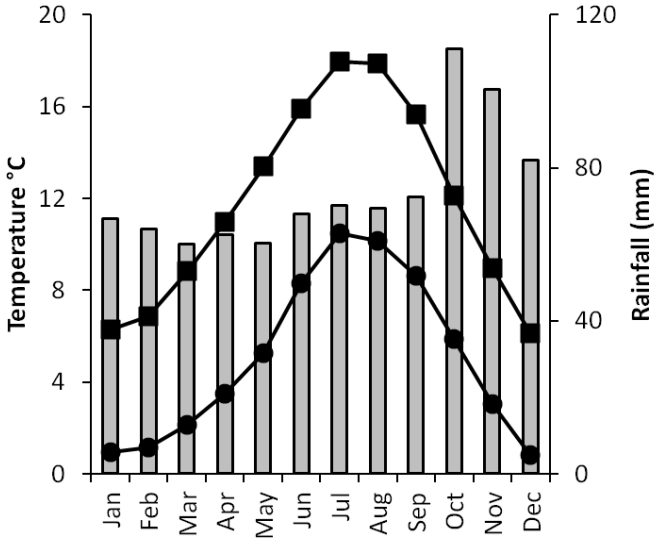

Figure 1. Mean monthly maximum (squares) and minimum (circles) air temperatures and mean total monthly rainfall recorded 0.5 km from the study site for 1991-2012.

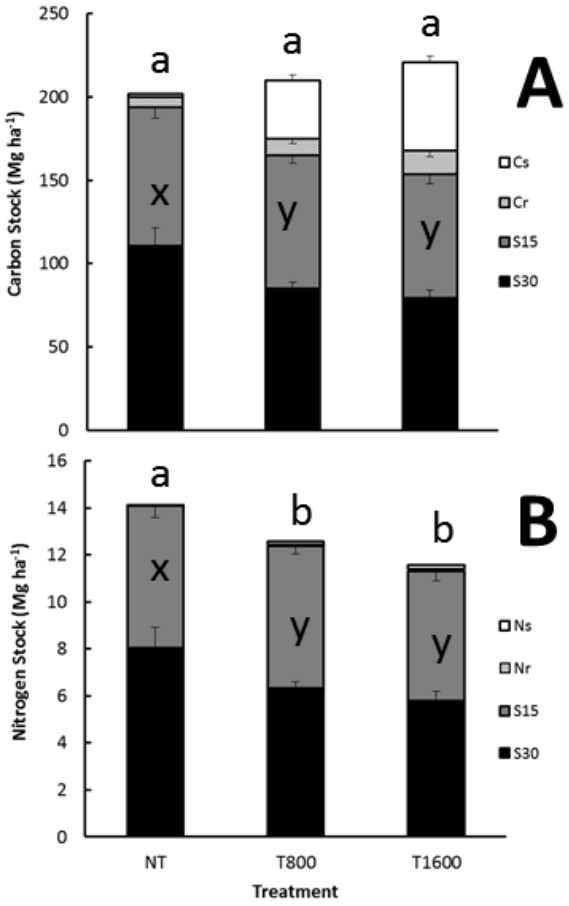

Figure 2. Partitioning of carbon (A) and nitrogen (B) between soil and plant fractions 21 years after land use change from arable to fallow (NT) or afforested (T). Fractions are soil 0 – 15 cm (S15), soil 15 – 30 cm (S30), plant roots (Cr & Nr) and plant aboveground (Cs & Ns). Vertical bars represent standard error of means, n = 9 (NT) and 18 (T). Similar letters no significant difference ($P \geq 0.05$) between treatments in either total (0-30 cm) soil stocks (x,y) or system (plant + soil) stocks (a,b).